Journal of Cleaner Production 216 (2019) 268–276

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

The significance of measuring embodied carbon dioxide equivalent in water sector infrastructure

Eleni Georgiou [a], Marie Raffin [b], Keith Colquhoun [b], Aiduan Borrion [a], Luiza C. Campos [a,*]

[a] Department of Civil, Environmental and Geomatic Engineering, University College London, Gower St, Bloomsbury, London, WC1E 6BT, United Kingdom
[b] Thames Water Utilities Ltd, Clearwater Court, Vastern Road, Reading, RG1 8DB, United Kingdom

ARTICLE INFO

Article history:
Received 26 September 2018
Received in revised form
11 December 2018
Accepted 2 January 2019
Available online 4 January 2019

Keywords:
Carbon footprint
Water industry
Embodied carbon
LCA (life cycle assessment)
Industry practice
Supply chain

ABSTRACT

For the water sector to cope with rising populations and the anticipated impacts of climate change, increasing amounts of construction output are needed to build water-related infrastructure. Amidst emerging operational energy efficiencies and gradual grid decarbonisation, the relative impact and extent of embodied carbon dioxide equivalent (embodied CO_2e) effected from the construction and maintenance of water sector infrastructure is likely to rise. For practitioners in the water and construction sectors, there is a growing need to be able to understand and account for embodied CO_2e. However, the contribution of embodied CO_2e as part of the whole life cycle impacts of water related infrastructure is disputed in the current literature, and with only a handful of studies suggesting it is important its significance is not established. This work aims to explore this issue, and provide clarity. This paper shows the calculations involved to measure the embodied and operational CO_2e of Old Ford Water Recycling Plant, a small blackwater recycling treatment facility producing 574 m^3/day of reclaimed water. For the analyses, embodied carbon dioxide coefficients (ECCs) are used that were provided by the water operator Thames Water Utilities Ltd (TWU), and based on its supply chain, and datasets from Ecoinvent v3.1, a commercially available assessment tool. The final aggregated carbon footprint values calculated are 1.430 $kgCO_2e$ (TWU in-house analysis) and 1.566 $kgCO_2e$ (Ecoinvent) per cubic metre of recycled water. The results show that the contribution of embodied CO_2e is significant, making up 50.7% (TWU) and 77.3% (Ecoinvent) of the total carbon footprint value in each analysis. The research identified that assessments could be improved if there was higher-quality data provided by manufacturers and suppliers on the embodied CO_2e content of materials, components, and equipment. This paper further illustrates differences between calculations using generic data (Ecoinvent) and supply chain data, and the difficulties involved in producing functionally equivalent life cycle inventories.

1. Introduction

Construction sector activities make up 30% of global anthropogenic greenhouse gas (GHG) emissions, and despite efforts the International Energy Agency (IEA) estimates that they will double by 2050 (IEA, 2014, 2016). For the water sector this is of particular importance; with the need for potentially significant amounts of construction output and energy needed to cope with rising populations, climate change and increasing service resilience, the provision of water and wastewater treatment services could become more resource intensive. Coupled with emerging operational energy efficiencies and gradual grid decarbonisation, a rise in construction output means that the relative contribution of embodied carbon is likely to increase. Over the last few years there is evidence to suggest that embodied carbon dioxide equivalent [1] (embodied CO_2e) associated with construction can make up for a significant proportion of the whole life cycle impacts in water and wastewater treatment (Mo et al., 2010; Singh and Kansal, 2018).

Previous life cycle assessment (LCA) studies in water and

* Corresponding author.
E-mail addresses: Eleni.Georgiou.09@ucl.ac.uk (E. Georgiou), Marie.Raffin@thameswater.co.uk (M. Raffin), Keith.Colquhoun@thameswater.co.uk (K. Colquhoun), a.borrion@ucl.ac.uk (A. Borrion), l.campos@ucl.ac.uk (L.C. Campos).

[1] Embodied CO_2e is a shorthand for embodied GHG emissions associated with the physical construction and maintenance of infrastructure (including material extraction and manufacture, asset replacement, assembly, deconstruction, and waste disposal). In this paper the terms 'embodied carbon' and 'embodied impacts' are also used to refer to embodied CO_2e.

https://doi.org/10.1016/j.jclepro.2019.01.014
0959-6526/

wastewater systems have mainly addressed GHG emissions from energy use (operational CO_2e or 'operational carbon') (Rothausen and Conway, 2011; Chang et al., 2017; Amores et al., 2013). A recent review on existing LCA studies on water and wastewater treatment systems by Byrne et al. (2017) found that only around half of all studies considered embodied CO_2e due to system construction. Of these studies, the majority concluded that embodied carbon is negligible compared to operational carbon (Igos et al., 2014; Bonton et al., 2012); however, a few papers have suggested it is important with embodied CO_2e effected from system construction and chemical use accounting for 30–40% of the whole life cycle impacts (Mo et al., 2010; Stokes and Horvath, 2010a; 2010b).

Both in the United Kingdom and internationally, embodied CO_2e accounting is becoming a component in tender selection, and its reduction is being linked to more direct cost saving and cost avoidance (Elhag, 2015). Organisations are increasingly becoming aware of its significance both within the supply chain as well as beyond the conventional boundary of capital investment e.g. by considering natural capital valuation and ecosystem services in the wider catchment to identify sustainability solutions (Battle, 2014; Smyth et al., 2017; UKGBC, 2017). In the UK, the introduction of carbon reporting of operational GHG emissions for large industry has given impetus to collaborative research in the water sector to develop carbon footprint methods to assess both operational and embodied impacts (Strutt et al., 2008). Water operators in the UK have already begun developing their own in-house bespoke embodied CO_2e calculation tools, using metrics and boundaries deemed most appropriate to their function. However, a barrier to the water sector's efforts to understand and account accurately for its embodied impacts is the lack of accessible and robust data from manufacturers and suppliers on the embodied CO_2e assessment of construction materials and products (Smyth et al., 2017; Gavotsis and Moncaster, 2015).

Several studies report that differences in this data can lead to a wide variation in the results of embodied CO_2e calculations for civil works (Dixit et al., 2010; Clark, 2013; De Wolf et al., 2016). Pomponi and Moncaster (2018) found that there is significant variability in the embodied carbon dioxide coefficients (ECCs) of common construction materials, which cannot be easily linked to a specific context (e.g. geographical location of production, specific manufacturing processes etc.). Sinha et al. (2016) and Herrmann and Moltesen (2015), who compared data from various LCA software including SimaPro and Gabi, found notable differences in the results obtained large enough to influence the conclusions. Differences in manufacturers' embodied carbon accounting approach (e.g. on which life cycle stages to include), and a lack of available information to formulate ECCs are the issues mainly responsible for this data variability (De Wolf et al., 2016). De Wolf et al. (2017) discussed current industry approaches in embodied CO_2e measurement through a detailed literature review, and through interviews with industry practitioners. Results showed that there is a demand for reliable product life cycle inventories to be provided by manufacturers and suppliers.

The significance of the contribution of embodied CO_2e as part of the whole life cycle impacts of water related infrastructure is unclear in the current literature. The aim of the present work is to address this issue, and provide clarity. In this paper, the embodied and operational CO_2e of Old Ford Water Recycling Plant, a blackwater recycling demonstration plant, are calculated using embodied carbon data (ECCs) that were directly provided by the water operator Thames Water Utilities Ltd (TWU), and ECCs from Ecoinvent v3.1, a commercially available assessment tool. This paper gives a first-hand insight into the challenges involved in the carbon footprint calculation of a blackwater recycling demonstration plant.

It is important to note that whilst this work focused retrospectively on the embodied CO_2e associated with a single piece of wastewater-related infrastructure, it has become clear that in the business decision making process embodied CO_2e should be considered in the round alongside cost, affordability, water quality and other challenges to inform resilient and more sustainable business decisions in the water sector and beyond.

2. Description of the site: Old Ford Water Recycling Plant

Old Ford Water Recycling Plant (OFWRP) is located in London, South East England, an area classified as 'seriously water stressed' by the national authorities (Environment Agency and Natural Resources Wales, 2003). The plant mines and treats raw sewage to supply non-potable water to the Queen Elizabeth Olympic Park for toilet flushing and irrigation via a dedicated non-potable reuse (NPR) network. The plant was set up as part of the Olympic Delivery Authority's sustainable water strategy to achieve a 40% reduction in potable water consumption. It also serves as a research facility to help inform possible future capital investments.

The plant is designed to supply a maximum flow of 574 m^3 per day of non-potable water. First, raw sewage is treated successively through underground septic tanks and 1 mm screens to eliminate grit and screening that could potentially damage downstream processes. The pre-treated sewage then feeds into the membrane bioreactor (MBR) where parts of organic, inorganic and microbiological compounds are removed. The MBR itself is a combination of an activated sludge process, consisting of an anoxic and an aerobic zone, and an ultrafiltration membrane process (UF) with a nominal pore size of 0.04 μm. Polyaluminium chloride is added to the return activated sludge of the MBR to precipitate phosphorus compounds. The granular activated carbon (GAC) process is used as a polishing step to remove any remaining colour in the MBR effluent. The reclaimed water is finally disinfected with sodium hypochlorite to achieve a chlorine residual of between 0.3 and 1.5 mg/L before being supplied to the NPR network. Odour control is carried out using two activated carbon filters.

3. Methodology

3.1. System boundaries and scope

The total aggregated carbon footprint (CF) of OFWRP associated with the life cycle inventory (LCI) in each of the analyses is normalised over a 25-year operation lifetime to the production of 1 m^3 of recycled water, which is defined as the functional unit. The construction and operation life cycle phases are considered, whilst emissions relating to dismantling and asset disposal are excluded (see Fig. 1). This system boundary was chosen to minimise uncertainty and concentrate on emissions associated with capital construction processes and the operation phase. The construction phase includes terrain excavation, production of building materials, plant equipment, chemical production, and materials replacement. The operational phase accounts for the production of grid electricity. Although renewably self-generated electricity produced at other sites run by the local water operator satisfies 20% of its demand on the whole, this has not been applied to this study as OFWRP does not produce any onsite power. In both phases, the transport of material inputs to the plant site is accounted for. The impact of sludge treatment and disposal of waste sludge from the septic tanks and the MBR is also accounted for as part of the operational phase. Fugitive emissions from the septic tanks and the MBR however are a function of the biology and chemistry of processes; these are omitted from this study seeing as they remain unchanged and unaffected as a result of any proposed investment

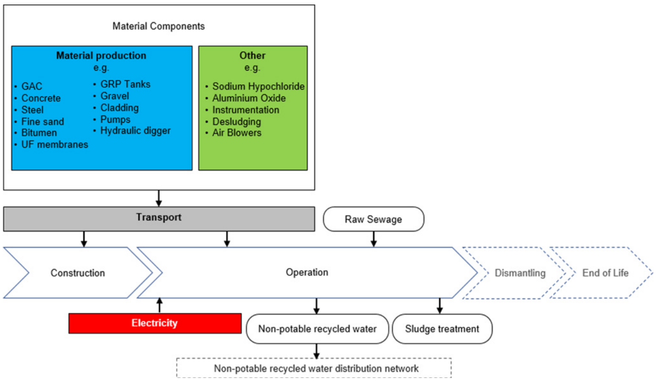

Fig. 1. OFWRP LCI system boundaries. Emissions relating to plant construction and operation are included, whilst dismantling, asset disposal (End of Life), and the construction of the non-potable recycled water distribution network are excluded from the scope of this study. The production and transport of building materials, components, and consumables (for chemical consumption) are accounted for, and the impact of grid electricity production as well as sludge treatment is assessed for a 25-year operational phase.

to reduce carbon intensity in the water infrastructure itself unless specific measures are taken to reduce them (UKWIR, 2012).

The life cycle of building materials, steel components, pipes and tanks, was assumed to be 50 years whereas the service life of plant equipment (compressors, air blowers, filter receivers etc.) was assumed to be 25 years. Pumps, instrumentation and UF membranes were assumed to have a service life of 10, 5 and 7 years respectively, and replacements were accounted for accordingly.

3.2. Life cycle and impact assessment methodologies

Primary industry data was collected from Thames Water Utilities Ltd. ('the water operator') to carry out an LCI analysis of the capital assets and operational requirements of OFWRP ('TWU LCI analysis'). The list of equipment in the site operating manuals and the engineering drawings for civil works were both used to produce a detailed bill of quantities. The ECCs used to calculate the embodied and operational CO_2e were taken from the inventory of emission factors provided by the water operator, which is compiled and updated annually (Table 1). ECCs from the Ecoinvent database (v3.1) were then obtained to reproduce the various component input flows in a separate LCI (Table 2). The ReCiPe midpoint hierarchist indicator approach was used for the life cycle impact assessment (LCIA). Material inputs having an exact or equivalent counterpart in the two LCIs are analysed explicitly. The datasets for which no counterparts could be identified are categorised in 'other material component production' in Tables 1 and 2, and 'other' in graphs. Descriptions and hypotheses for key inputs in the LCIs are outlined below:

3.3. Transport

For the purposes of normalisation, default settings for transport modes and distances considered by Borken-Kleefeld and Weidema (2013) were assumed in both LCIs for materials, components and equipment. In the absence of data from the manufacturing supply chain, sourcing and production of construction materials and products were assumed to take place in Europe. The transport distances by heavy goods vehicles ('32 tonne trucks') assumed to be involved in the production and sourcing of ferrous metals, plastics, and bitumen were 100 km, and 50 km for concrete and minerals. Similarly, transport distances by rail for ferrous metals and plastics were assumed to be 200 km and 600 km for bitumen. For the sourcing and production of GAC filter media, transport distances by truck and rail were assumed to be 50 km and 400 km respectively. The consumption of diesel fuel and electricity in haulage was included in both LCIs. Ecoinvent datasets additionally cite production of trucks, locomotives, wagons, rail infrastructure and vehicle disposal in land freight transport. For the TWU LCI analysis, the ECCs for rail and truck materials transport were drawn from the industry recommended UK government guidelines for company reporting (BEIS et al., 2017).

3.4. Sludge treatment

The daily volume of waste sludge produced at OFWRP, averaged at 3 m^3, is fed into the Olympic Park primary sewer which connects to the major gravity sewer running to Beckton sewage treatment works (STW) in East London; waste sludge at Beckton STW is treated using anaerobic digestion. The volume of sewage gas (biogas from sewage sludge digestion) produced per cubic metre of sludge was estimated using data from Amiri et al. (2015) and Daelman et al. (2012). For the Ecoinvent LCI, sludge treatment and disposal are accounted for and the dataset is categorised in 'other'. The model represents normalised methane emissions from the overall off-gas leaving the water and sludge lines, raw sludge settling, thickening and digestion, and onsite gas storage. The model includes transport of the digested and dewatered sludge to the incineration facility.

3.5. Electricity

The total electricity consumption, averaged at 1.766 kWh per m^3 of recycled water delivered to the network, was derived from plant records. Grid electricity production in the LCIs is modelled at medium voltage level assuming the UK electricity consumption profile. Ecoinvent accounts for the direct emissions but excludes transmission losses, and indirect emissions relating to the construction of transformer stations and the distribution grid.

For the TWU LCI the industry recommended 1-year grid average ECC was used in accordance with the 2017 UK government BEIS guidelines, accounting for direct and indirect GHG emissions (BEIS et al., 2017). Direct emissions include the production of grid electricity at UK power plants, the transmission network and losses, whilst indirect emissions account for the fuel production upstream.

3.6. Pumps

For the Ecoinvent LCI, the production of pumps was modelled as their weight (based on manufacturer datasheets) assuming they consist of pure stainless steel. No other processes were considered aside from chromium steel production and hot rolling. It was assumed the pumps are manufactured in Europe using a European production mix (Frischknecht et al., 2005; Weidema et al., 2013). For the TWU LCI, the datasets for pumps were drawn from the manufacturer and subdivided into 'centrifugal non-submersible' and 'wet well submersible' (Xylem, 2013a, 2013b; 2013c).

3.7. Ultrafiltration membranes

No specific Ecoinvent dataset was found to model polyvinylidene fluoride production, which is the material used for UF membranes. Instead, the model for polyvinyl chloride production was used as a proxy being the closest dataset found in the database. The membranes were modelled as their weight, and the steel casing for the cells was assessed separately. For the TWU LCI the UF membranes were modelled as part of the 'MBR plant' aggregate ECC (Table 1).

Table 1
Life cycle inventory of OFWRP with ECCs used by Thames Water on the basis of 1 cubic metre of water recycled ('TWU LCI'), Reference year 2017.

Component	Quantity [unit]	ECCs $kgCO_2e$/unit	Assumptions
ELECTRICITY	1.766 kWh	0.38443	Direct and indirect GHG emissions from UK grid electricity production, transmission and distribution (BEIS et al., 2017).
TRANSPORT -			
Materials transport by truck	0.154 tkm[a]	0.14048	Direct emissions from truck fuel consumption (BEIS et al., 2017).
Materials transport by rail	0.107 tkm	0.03394	Rail freight. Direct emissions from diesel and electric rail (BEIS et al., 2017).
MATERIAL PRODUCTION			
Concrete	4.68E-04 m^3	814.89	
Bulk earthworks	7.666E-04 m^3	2.175	Terrain excavation for foundations. Fuel consumption of the crawler tractor and hydraulic digger included in the model.
GAC plant	4.77E-06 m^3/hr	2689.446	Complete GAC plant, pressure vessels, interconnecting pipework and valves. Excludes GAC/sand washing plant.
Structural metalwork -			
Access platform	1.214E-05 m^2	334.116	
Hand railing	5.728E-05 m	214.938	
Cladding	2.132E-04 m^2	345.848	Roofing, building elevation, tank cladding and wire mesh flooring. Assumed to be made of steel.
Structural steel	9.00E-05 m^2	356.21	Complete above ground steel superstructure. Excludes building services and process specific equipment.
Steel components -			
Piles	5.025E-05 m^2	2.362	Piling for temporary foundation shaft.
Pressure vessels	2.88E-06 m^3	1242.545	Plate steel sections, above ground pipework, fixture and fittings. Based on volume capacity.
Stainless steel components	3.468E-05 m^2	523.405	Dry stainless steel weight.
Tanks	1.339E-04 m^3	63.027	Glass-fused-to-steel tanks production based on volume capacity. Inclusive of epoxy resin application.
Overhead travelling crane & gantry	3.818E-07 T	1521.511	Heavy duty electric chain hoist + crane beams.
Chemical dosing skids	5.23E-06 T	345.848	Filtrate dry skid steelwork.
Component	**Quantity [unit]**	**ECCs $kgCO_2e$/unit**	**Assumptions**
MBR plant	4.96E-06 m^3/hr	2872.397	Membrane filters, MEICA[b] equipment. Excludes building superstructure and high-pressure pumps.
Pumps -			
Wet well pumps	1.867E-05 kW	480.989	Production of ITT Flygt submersible pumps. Environmental product declarations (Xylem, 2013a, 2013b; 2013c).
Centrifugal non-submersible pumps	5.489E-05 kW	58.979	
Dosing equipment	3.445E-06 m^3	610.57	
Pipes -			
Steel pipes	2.770E-05 m	82.80	Extrusion, bituminous primer coating and fibreglass wrapping included.
uPVC pipes	9.289E-05 m	5.20	uPVC polymer production and pipe extrusion, laid with minimal bedding surround.
GRP[c] -			
Chemical dosing kiosk	1.406E-06 m^2	7.093	
GRP tanks	6.122E-05 m^3	109.926	Volume input refers to liquid volume the tanks can hold and not their material quantity. Plant records.
MBR -			
Air handling unit/filter receiver	1.031E-04 m^3/hr	1.896	
Compressor	1.428E-04 m^3/day	3.012	
MBR air blowers	3.67E-05 m^3/hr	0.344	
OTHER MATERIAL COMPONENT PRODUCTION ('other' in graphs)			
Building services	1.80E-04 m^2	9.53628	Telemetry, SCADA hardware, and telecommunications equipment.
Bubble diffuser domes	9.758E-06 m^2	369.098	Steel sheets covering gross surface area domes in aeration tank.
Ion exchange/softener vessels	8.19E-08 m^3/hr	1099.650	
Air blowers	5.72E-05 m^3/hr	0.344	
Desludging	2.12E-07 m^3/hr	910.113	
Instrumentation	1.32E-03 items	94.709	Fittings e.g. valves, flow meters, temperature sensors, transmitters, etc.
Perforated screens	1.69E-05 m^3/hr	2.741	Production of screens based on processing capacity.
Ventilation system	8.59E-04 m^3/hr	2.304	Production of the odour control plant based on its processing capacity.
Odour control	2.880E-06 m^3	1242.55	

Notes.
[a] tkm or tonne-kilometre: a unit of freight equal to the transport of one metric ton of goods by 1 km.
[b] MEICA: Mechanical, Electrical, Instrumentation, Controls, Automation (industrial equipment systems).
[c] GRP: Glass reinforced plastic (or fibreglass); a polyester material reinforced with glass fibre.

3.8. Granular activated carbon

For the Ecoinvent LCI, to model GAC production used in pressure filters in the GAC and odour control plants, it was necessary to disaggregate the 'GAC plant' into the elements 'filter media' and 'pressure vessels' (analysed in 'steel'). The service life of GAC beds was noted to be two years based on plant datasheets, which was incorporated into the analyses to account for all bed refills throughout a 25-year operation period. For the Ecoinvent LCI, the ECC was drawn from Bayer et al. (2005) who modelled the main

Table 2
Life cycle inventory of OFWRP using Ecoinvent V3.1 ECCs on the basis of 1 cubic metre of water recycled.

Component	Quantity [unit]	ECCs $kgCO_2e$/unit	Assumptions
ELECTRICITY	1.766 kWh	0.183	Transformation of 1 kWh of high to medium voltage electricity assuming the UK electricity consumption profile. Imported electricity is accounted for. The infrastructure itself and losses incurred are excluded.
TRANSPORT -			
Materials transport by 32 tonne truck	0.154 tkm	0.171	
Materials transport by rail	0.107 tkm	0.0608	
MATERIAL COMPONENT PRODUCTION			
Concrete	4.68E-04 m^3	285	Production of ready-mixed concrete based on Swiss production mix and electricity.
Bulk earthworks	7.666E-04 m^3	0.550	Terrain excavation. Assumed to be dug with a hydraulic digger. The model includes production of the machine and fuel consumption.
GAC -			
Filter media	0.017 kg	11	Virgin GAC production assumed (Bayer et al., 2005).
Odour control	0.029 kg	11	Production of virgin GAC for odour control (Bayer et al., 2005; Alfonsín et al., 2015). Filter casing and fans excluded.
Structural metalwork -			
Access platform & hand railing	6.94E-05 m^2	198	Assumed to be made from aluminium cladding.
Cladding	2.132E-04 m^2	198	Roofing, tank cladding and wire mesh flooring. Production of aluminium wire mesh.
Structural steel	0.022 kg	2.500	Rebar and steel superstructure. Inclusive of low and unalloyed-steel production, and hot rolling.
Steel components -			
Carbon steel	9.957E-04 kg	2.260	Production of chemical dosing skids, the overhead travelling crane and lifting gantry. The model includes raw material extraction, unalloyed steel production and hot rolling.
Pressure vessels	1.527E-03 kg	2.290	Low carbon structural steel. Production of a mix of differently produced steels, and hot rolling.
Component	Quantity [unit]	ECCs $kgCO_2e$/unit	Assumptions
Stainless steel components	0.059 kg	5.010	Production of pumps, ultrafiltration cells, tanks and other components. Chromium steel production and hot rolling was included assuming a density of 8000 kg/m^3 based on stainless steel type 316.
Pumps	0.001846 kg	5.010	See entry on 'stainless steel components'.
Pipes -			
Steel pipes	7.998E-04 kg	5.240	Proxy dataset used to model chromium steel pipe fabrication based on global production.
uPVC pipes	6.080E-05 kg	3.230	The model includes production of PVC polymer, transport, pipe extrusion and packaging.
GRP -			
Enamelling	7.615E-05 m^2	13.100	Vitreous porcelain enamelling used for coating tanks. The model includes all major process stages and the inventory for the materials used in enamelling.
GRP tanks	2.052E-03 kg	8.960	Gate to gate inventory for the injection moulding of glass fibre with polyamide resin, plus material inputs, and the infrastructure. Calculated from the volume of GRP used and a density of 1740 kg/m^3.
MBR -			
UF membranes	6.965E-04 kg	4.97	Polyvinyl chloride used as a proxy dataset for PVDF. PVC polymer production (suspension polymerisation) and extrusion to obtain foils.
OTHER MATERIAL COMPONENT PRODUCTION ('other' in graphs)			
Sludge treatment	5.23E-03 m^3	0.45055	The proportion of biogas volume produced is assumed per sludge volume treated.
Artificial gravel	9.94E-03 kg	0.00492	Land preparation. The model includes extraction and crushing (Muñoz et al., 2008).
Fine sand	2.34E-03 kg	0.00441	Sand for civil works. The model includes extraction from mine.
Bitumen	4.68E-05 kg	0.587	Road paving. The model includes oil extraction, transport and the refinery process.
11% Poly aluminium chloride (PAC)	3.88E-06 kg	1.730	Aluminium oxide (Al_2O_3). Production of 11% poly aluminium chloride coagulant. Calculated based on a weight.
Sodium hypochlorite NaOCl 10%–15%	0.121 kg	0.958	Sodium hypochlorite, without water, in 15% solution state for disinfecting, clean-in-place and maintenance wash of the ultrafiltration cells.

production processes of a coal-derived type of GAC manufactured in Europe by Chemviron (Calgon, 1999). The ECC from their study was deemed appropriate as the physicochemical properties of the GAC modelled are similar to those of the GAC type used in the plant. It should be noted that the GAC used in OFWRP is derived from vegetal sources.

GAC requirements for the odour control plant were derived from the activated carbon filter modelled by Alfonsín et al. (2015). Distinct aggregate ECCs were used to model the 'GAC plant' and the 'odour control plant' for the TWU LCI, and both ECCs were sourced from water operator provided datasets (Table 1).

3.9. Steel

Material inputs for steel were categorised in 'steel components' and 'structural steel' (categorised into 'structural metalwork'). Ecoinvent datasets reflect steel production in plants located in Europe.

3.10. Concrete

For the Ecoinvent LCI, the dataset reflects typical concrete mixture in Switzerland using Swiss grid electricity. Steel rebar used in concrete foundations was assessed separately as 'structural steel'. For the TWU LCI, the ECC accounts for in-situ concrete production, steel rebar reinforcement and formwork.

4. Results and discussion

This section discusses the results holistically and evaluates important factors which can pose as limitations to the validity of the calculations. Fig. 2 reveals some interesting findings. The final total aggregated CF values calculated for OFWRP are 1.430 and 1.566 $kgCO_2e/m^3$ using ECCs from TWU (in-house) analysis and Ecoinvent respectively. It is important to note that the final figures are similar in magnitude. The embodied CO_2e comprises subgroup categories 'materials production' and 'other'; the embodied CO_2e figures calculated for the TWU and Ecoinvent analyses are 0.725

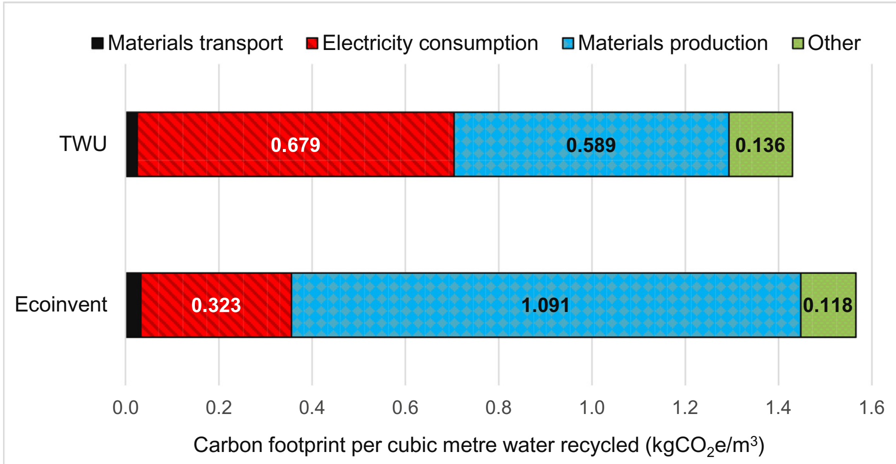

Fig. 2. Comparison and breakdown of subgroup contribution to the total OFWRP CF. The total aggregated CF values for the TWU and Ecoinvent analyses are 1.430 and 1.566 $kgCO_2e/m^3$ respectively. For the categorical grouping of components, see Tables 1 and 2

and 1.210 $kgCO_2e/m^3$ respectively. The findings show that in both analyses embodied CO_2e effected from the construction and maintenance of capital assets (normalised over a 25-year operation period per m^3 of water recycled) is a significant contributor to the whole life cycle impacts of OFWRP, making up 50.7% and 77.3% of the total CF using TWU and Ecoinvent data sets respectively. The contribution of operational CO_2e remains a significant part of the total CF assessments; the figures calculated for OFWRP are 0.679 and 0.323 $kgCO_2e/m^3$ in the TWU and Ecoinvent analyses respectively (making up 47.5% and 20.6% of the whole life cycle impacts in each case).

Fig. 3 shows the individual and relative contributions to the total CF of OFWRP calculated in both analyses. Apart from 'electricity' (operational CO_2e) being the single largest individual contribution

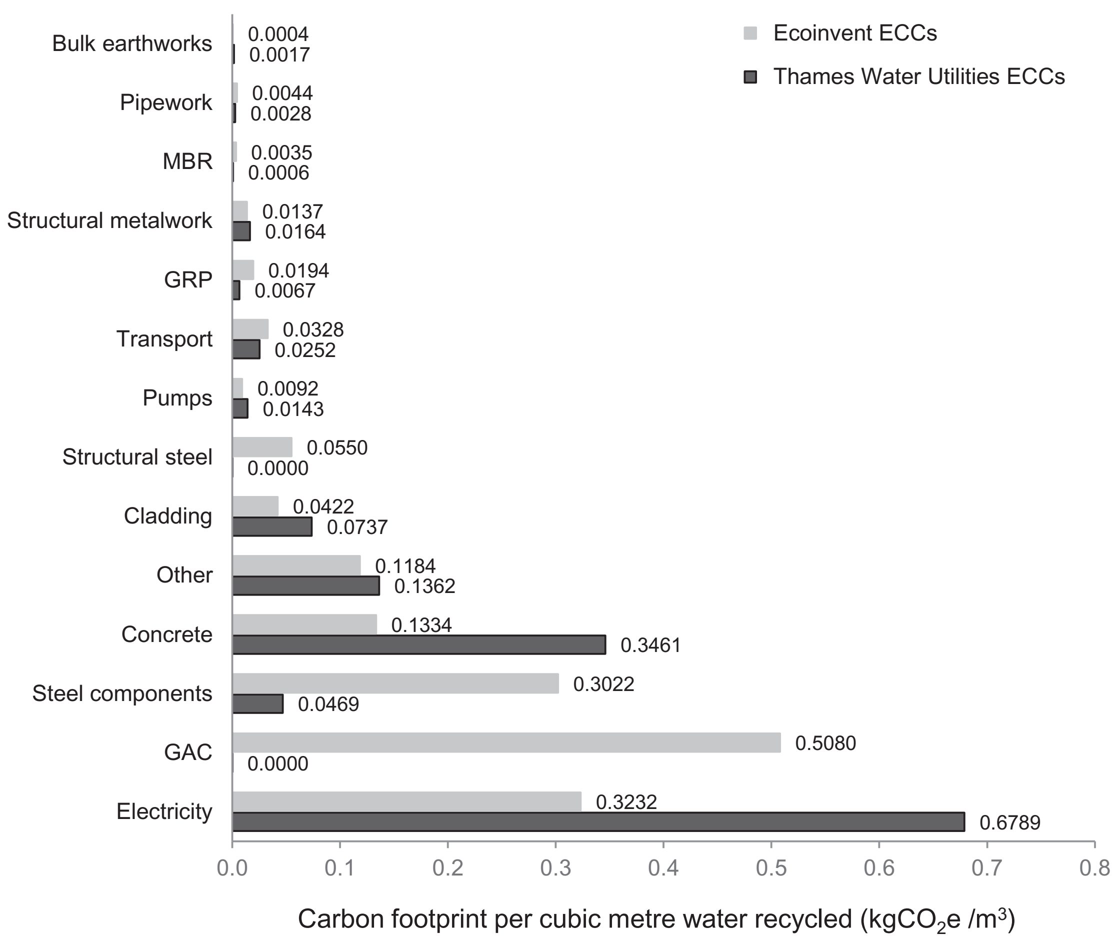

Fig. 3. Individual and relative contributions to the total OFWRP CF.

in Fig. 3, the rest of the noteworthy contributions are: GAC, concrete, steel (components and structural), cladding, and 'other'. The rest of the contributions shown in Fig. 3 are at least an order of magnitude smaller.

As shown in Figs. 2 and 3, there is a discrepancy between the operational CO_2e figures calculated in the TWU and Ecoinvent analyses. This difference can be attributed to the fact that Ecoinvent ECCs are based on generic secondary data sets which are not up to date nor specific to the UK electricity market. The Ecoinvent ECC is representative of British electricity production during the years 2010–2013, and relies on International Energy Agency (IEA) statistics to apportion the fuel mix for British electricity production (Itten et al., 2014). The ECC used for the TWU analysis on the other hand is specific to the UK national electricity grid supply and generation. It is more representative than the IEA data as it is derived from UK GHG Inventory data and the UK fuel consumption data, which are updated annually (Jones et al., 2017; BEIS et al., 2017). The Ecoinvent CF for electricity production is thusly a considerable underestimate. It should be noted that as the UK energy grid decarbonises gradually including decentralised self-generated renewable electricity substitutes, there will be a significant impact on the ECC applied. This will in turn considerably alter the aggregate CF values calculated for operational CO_2e. The carbon intensity of the electricity grid is generally on a downward trajectory, but it can be volatile (Li and Trutnevyte, 2017; Staffell, 2017; Green and Staffell, 2016); in this case, as far as the operational CO_2e calculations for OFWRP are concerned, possible future energy mix scenarios were not considered, and the potential future decarbonisation of the UK national electricity grid was not accounted for. The ECC published in government guidelines for company reporting (BEIS et al., 2017) was used instead to account for the assumed 25-year operation period of the plant in the TWU LCI analysis.

A few elements that are immediately evident from comparing the two analyses are the lack of functional equivalence and the different units in which quantities of products, materials and equipment are accounted for (Tables 1 and 2). For example, the component 'concrete' in the TWU LCI analysis includes steel reinforcement and formwork, whereas this is not the case with Ecoinvent where structural steel reinforcement and concrete are recorded separately. Several components were accounted for in different units e.g. 'structural steel', 'GRP tanks', 'pipework', 'MBR plant', 'pumps', etc. The lack of a standard method for data collection on the number, specification and various types and grades of components was one of the most significant and difficult issues to tackle in the context of this study. Commercial software such as the Ecoinvent database often do not contain ECCs readily available in a format that accounts for the complete supply chain of specific products and their subcomponents. For the bill of quantities data to be translated from the original OFWRP format of metrics in which it was collected, a few assumptions were made as detailed in Section. 3.2. It is also worth noting that comparative assessments involving structural and other building materials should be based on units of performance instead of units of mass (Pomponi and Moncaster, 2018). In other words, one material or product with a relatively higher embodied CO_2e compared to another may have a longer life service and better performance requiring fewer replacements. The environmental friendliness of one product or material over others should not simply be judged based on a lower embodied carbon footprint, and just during its manufacturing stage (Pomponi and Moncaster, 2018). There are other factors apart from a lower embodied carbon content that need to be taken into account when choosing a product or material such as the final disposal and waste degradation of materials and consumables. For example, due to the expected increase of membrane production and use alternative end of life options merit further study to limit the final disposal of membranes sent to landfill (Lawler et al., 2012, 2013).

Due to a lack of information regarding the exact nature of the manufacturing supply chain for material components, it was assumed that products and components were manufactured in and sourced from Europe. The default settings for transport modes and distances developed by Borken-Kleefeld and Weidema (2013) were applied in both LCIs (Tables 1 and 2). Moreover, whilst GAC used in OFWRP is produced using vegetal sources, due to poor availability of data, Ecoinvent data sets assumed production of virgin GAC based on hard coal. This is evidently a considerable overestimate of the actual impacts of GAC production which introduces a large uncertainty in the overall CF results.

5. Uncertainty analysis

In Section 3.2. input flow quantities in the LCI analysis (or 'components' in Tables 1 and 2) have been described in single figures (mean values). This practice involves uncertainty, which can be caused by various reasons as previously described by Frischknecht et al. (2005) cited in Muñoz and Fernández-Alba (2008): variability and stochastic error, appropriateness of the data, model uncertainty, and omitting or missing data. In this study, only the first two types of uncertainty have been considered at the inventory level, and the effect has been quantified on the overall CF results in each analysis. It must be noted that uncertainty due to the characterisation models used at the LCIA stage is considered as integral as the uncertainty relating to inventory analysis. Uncertainty due to the characterisation models has not been included in this analysis as these models do not reveal this information as of yet.

5.1. Stochastic Monte Carlo simulation model

The method followed to quantify the overall uncertainty related to the CF results involves two steps. Firstly, the uncertainty of individual input flows for electricity, transport and material production 'components' (as shown in Tables 1 and 2) was determined and then applied to the overall CF results in each analysis using a stochastic Monte Carlo simulation model. The uncertainty relating to the amount of a specific input flow cannot often be derived since the information is provided in a single value originating from a single source, without any further information (Frischknecht et al., 2004). The uncertainty related to individual input flows was not available in either analysis; uncertainty factors first had to be assigned to each of the inventory component quantities shown in Tables 1 and 2 In order to do so, the simplified approach developed by Frischknecht et al. (2004) was applied. In this standard procedure a lognormal distribution of uncertainty is considered, and the squared geometric standard deviation (σ^2) is used to express uncertainty factor contributions. Attribution of uncertainty factors to each of the inventory components in each analysis was carried out by means of a semiquantitative approach which included expert judgement and a data quality assessment using indicators from a pedigree matrix developed by Weidema and Wesnaes (1996) cited in Frischknecht et al. (2004) and Muñoz and Fernández-Alba (2008). Table 3 gives a summary of the uncertainty factors (expressed as 'squared geometric standard deviation' (σ^2)) assigned to the components in each analysis.

In order to establish a minimum-maximum uncertainty range in each case, the stochastic Monte Carlo simulation model was used, and the entire inventory was ran 10 000 times at a confidence interval of 95% for each analysis. This allowed for a statistically adequate and sturdy number of results. The principle followed in the Monte Carlo simulation is that singular estimates are taken as

Table 3
Uncertainty factors estimated for inventory components (expressed as 'squared geometric standard deviation').

Squared geometric standard deviation (σ^2)	TWU LCI components	Ecoinvent LCI components
1.1	Electricity, GAC plant, Access platform, Hang railing, Cladding, Odour control, Pressure vessels, Wet well pumps, Centrifugal non-submersible pumps, Dosing equipment, Air handling unit/filter receiver, Compressor, MBR air blowers, Bubble diffuser domes, Ion exchange/softener vessels, Air blowers, Desludging, Instrumentation, Perforated screens	Access platform & hand railing, Cladding, Structural steel, Carbon steel, Pressure vessels, Stainless steel components, Steel pipes, uPVC pipes, Enamelling, GRP tanks, Artificial gravel, Fine sand, Bitumen, 11% Poly aluminium chloride PAC, Sodium hypochlorite NaOCl 10 %–11%, Sludge treatment
1.3	Structural steel	
1.4		Pumps
2.0		Bulk earthworks
2.1	Bulk earthworks, Ventilation system	Materials transport by 32 tonne truck, Materials transport by rail
2.4	Materials transport by truck, Materials transport by rail	UF membranes
2.6	Chemical dosing skids, MBR plant, Chemical dosing kiosk, GRP tanks	Electricity
3.0	Concrete	
3.1	Overhead travelling crane & gantry, Steel pipes, uPVC pipes, Piles	
3.3	Tanks	
3.4	Stainless steel components	
4.2	Building services	Concrete, Filter media, Odour control

Table 4
Results of the uncertainty analysis with the stochastic Monte Carlo simulation model with a 95% confidence interval and 10 000 simulations.

TWU CF Uncertainty Range	Ecoinvent CF Uncertainty Range
1.03–2.09 $kgCO_2e/m^3$	1.21–2.61 $kgCO_2e/m^3$

mean estimates which are then replaced with random variables drawn from probability density functions (Muñoz and Fernández-Alba, 2008; La Grega et al., 1994). Each time the simulation was ran new variables were selected based on the uncertainty factor assigned to each of the components thusly creating a random set of numbers to obtain a new score. Table 3 gives a summary of the uncertainty factors assigned to the components in each analysis. For example, a σ^2 of 1.1 for electricity in the TWU LCI analysis indicates that a mean value of 1.766 kWh/m^3 has minimum maximum values of 1.30–2.41 kWh/m^3. For the Ecoinvent LCI analysis, the GAC ('filter media'), 'odour control', 'concrete' and 'electricity' components display a high level of uncertainty. For the TWU LCI analysis, the higher uncertainty is seen in steel components and concrete.

5.2. Results of the uncertainty analysis

After running the Monte Carlo simulation model 10 000 times, with a confidence interval set at 95%, the results displayed in Table 4 are obtained. It can be deduced that the results are uncertain as there is a wide variability in the minimum – maximum ranges obtained. The difference between the minimum and maximum values is approximately twofold in both analyses. However, a lower difference and a stricter uncertainty range can be observed in Table 4 for the TWU LCI analysis compared to the score for the Ecoinvent LCI analysis. Moreover, an overlapping of their distributions can also be seen. With the production stage of material components and products being at the remit of the construction supply chain, the variability in these uncertainty ranges can be considerably reduced through the provision of accurate, up to date and representative data (for the material or product analysed) from suppliers and manufacturers.

6. Conclusions

The results of this study show that the contribution of embodied CO_2e effected from the construction and maintenance of capital assets can potentially be significant in the whole life cycle impacts of the water recycling plant. By providing one industrial perspective on the carbon footprint assessment of a water recycling plant this work underlines the potential importance of accounting for embodied CO_2e in project design, delivery and operation.

The difference between the two calculated results reveals that there is a clear need for robust and accessible ECCs to be made available from the supply chain; this would aid in reducing differences in the measurement of embodied impacts which would improve the accuracy of LCAs carried out by practitioners. More to this point, the sizeable discrepancy between the operational CO_2e figures accounting for British electricity production in the two analyses provides clear evidence that the use of generic data can distort the true extent of embodied impacts.

More efforts in data collection and data quality assessment are still required to produce databases for reliable and transparent product LCIs and ECCs in the appropriate format which would better inform industry practitioners in design decision making. The creation of environmental product declarations (EPDs) provides an opportunity for practitioners to obtain robust, representative and up to date data directly from manufacturers and suppliers. The manufacturing industry should be encouraged to improve availability of robust assessments of embodied CO_2e for construction products and materials as well as improving data collection methods. The use of geographically and technologically appropriate embodied CO_2e information would help to reduce variability in assessments.

Disclaimer

The views and opinions expressed in this paper are those of the authors and are the product of ongoing research. They do not represent the position or opinions of either UCL or Thames Water Utilities Ltd.

Acknowledgements

The authors would like to acknowledge the close collaboration of Thames Water Utilities Ltd.

References

Alfonsín, C., Lebrero, R., Estrada, J.M., Muñoz, R., Kraakman, N.B., Feijoo, G.,

Moreira, M.T., 2015. Selection of odour removal technologies in wastewater treatment plants: a guideline based on Life Cycle Assessment. J. Environ. Manag. 149, 77–84.
Amiri, A., Tsutsumi, J., Nakamatsu, R., 2015. Energy from domestic wastewater and recovering the potential energy of sewage sludge. J. Energy Technol. Pol. 5 (12).
Amores, M.J., Meneses, M., Pasqualino, J., Antón, A., Castells, F., 2013. Environmental assessment of urban water cycle on Mediterranean conditions by LCA approach. J. Clean. Prod. 43, 84–92.
Battle, G., 2014. Embodied carbon industry task force recommendations. Proposals for standardised measurement method and recommendations for zero carbon building regulations and allowable solutions [ONLINE] Available at: https://www.ukgbc.org/wp-content/uploads/2017/09/Delivering-Low-Carbon-Infrastructure.pdf.
Bayer, P., Heuer, E., Karl, U., Finkel, M., 2005. Economical and ecological comparison of granular activated carbon (GAC) adsorber refill strategies. Water Res. 39 (9), 1719–1728.
Bonton, A., Bouchard, C., Barbeau, B., Jedrzejak, S., 2012. Comparative life cycle assessment of water treatment plants. Desalination 284, 42–54.
Borken-Kleefeld, J., Weidema, B.P., 2013. Global default data for freight transport per product group. Manuscript for special ecoinvent, 3.
BEIS, Hill, N., Bramwell, R., Harris, B., 2017. 2017 Government GHG Conversion Factors for Company Reporting: Methodology Paper for Emission Factors - Final Report. Department for Business, Energy & Industrial Energy, UK. Available at: https://www.gov.uk/government/publications/greenhouse-gas-reporting-conversion-factors-2017, 26 November 2017.
Byrne, D.M., Lohman, H.A., Cook, S.M., Peters, G.M., Guest, J.S., 2017. Life cycle assessment (LCA) of urban water infrastructure: emerging approaches to balance objectives and inform comprehensive decision-making. Environ. Sci.: Water Res. Technol. 3 (6), 1002–1014.
Calgon, 1999. FILTRASORB® 300 & 400. [ONLINE] Available at: http://www.reskem.com/wp-uploads/2015/04/calgon-filtrasorbf300potable.pdf. (Accessed 7 January 2019).
Chang, J., Lee, W., Yoon, S., 2017. Energy consumptions and associated greenhouse gas emissions in operation phases of urban water reuse systems in Korea. J. Clean. Prod. 141, 728–736.
Clark, D.H., 2013. What Colour is Your Building?: Measuring and Reducing the Energy and Carbon Footprint of Buildings. RIBA.
Daelman, M.R., van Voorthuizen, E.M., van Dongen, U.G., Volcke, E.I., van Loosdrecht, M.C., 2012. Methane emission during municipal wastewater treatment. Water Res. 46 (11), 3657–3670.
De Wolf, C., Pomponi, F., Moncaster, A., 2017. Measuring embodied carbon dioxide equivalent of buildings: a review and critique of current industry practice. Energy Build. 140, 68–80.
De Wolf, C., Yang, F., Cox, D., Charlson, A., Hattan, A.S., Ochsendorf, J., 2016. Material quantities and embodied carbon dioxide in structures. No.4. In: Proceedings of the Institution of Civil Engineers-engineering Sustainability, vol. 169. Thomas Telford Ltd, pp. 150–161.
Dixit, M.K., Fernández-Solís, J.L., Lavy, S., Culp, C.H., 2010. Identification of parameters for embodied energy measurement: a literature review. Energy Build. 42 (8), 1238–1247 [online]. Available at: https://doi.org/10.1016/j.enbuild.2010.02.016.
Elhag, H., 2015. The 'carbon footprint' of sewer pipes: risks of inconsistency. No. 1. In: Proceedings of the Institution of Civil Engineers-engineering Sustainability, vol. 168. Thomas Telford Ltd, pp. 38–48.
Environment Agency, Natural Resources Wales, 2003. Water stressed areas – final classification. [ONLINE] Available at: https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/244333/water-stressed-classification-2013.pdf. (Accessed 8 January 2019).
Frischknecht, R., Jungbluth, N., Althaus, H.J., Doka, G., Dones, R., Heck, T., Hellweg, S., Hischier, R., Nemecek, T., Rebitzer, G., Spielmann, M., 2005. The ecoinvent database: overview and methodological framework (7 pp). Int. J. Life Cycle Assess. 10 (1), 3–9.
Gavotsis, E., Moncaster, A., 2015. Improved embodied energy and carbon accounting: recommendations for industry and policy. Athens J. Technol. Eng. 9–23.
Green, R., Staffell, I., 2016. Electricity in Europe: exiting fossil fuels? Oxf. Rev. Econ. Pol. 32 (2), 282–303.
Herrmann, I.T., Moltesen, A., 2015. Does it matter which Life Cycle Assessment (LCA) tool you choose? – a comparative assessment of SimaPro and GaBi. J. Clean. Prod. 86, 163–169.
IEA, 2014. Technology Roadmap. Energy Efficient Building Envelopes. International Energy Agency, Paris, France.
IEA, 2016. Annex 57 Evaluation of Embodied Energy & Embodied GHG Emissions for Building Construction - Report of Subtask 4: Recommendations for the Reduction of Embodied Carbon and Energy from Buildings. International Energy Agency.
Igos, E., Dalle, A., Tiruta-Barna, L., Benetto, E., Baudin, I., Mery, Y., 2014. Life Cycle Assessment of water treatment: what is the contribution of infrastructure and operation at unit process level? J. Clean. Prod. 65, 424–431.
Itten, R., Frischknecht, R., Stucki, M., 2014. Life Cycle Inventories of Electricity Mixes and Grid, Version 1.3. treeze Ltd., Uster, Switzerland.
Jones, L., Passant, N., Thomson, A., Wakeling, D., Buys, G., Forden, S., Gilhespy, S., Glendining, M., Gluckman, R., Henshall, P., Hobson, M., 2017. UK Greenhouse Gas Inventory, 1990 to 2015.
La Grega, M.D., Buckingham, P.L., Evans, J.C., 1994. Stabilisation and solidification. Hazard. Waste Manag. 641–704.
Lawler, W., Antony, A., Cran, M., Duke, M., Leslie, G., Le-Clech, P., 2013. Production and characterisation of UF membranes by chemical conversion of used RO membranes. J. Membr. Sci. 447, 203–211.
Lawler, W., Bradford-Hartke, Z., Cran, M.J., Duke, M., Leslie, G., Ladewig, B.P., Le-Clech, P., 2012. Towards new opportunities for reuse, recycling and disposal of used reverse osmosis membranes. Desalination 299, 103–112.
Li, F.G., Trutnevyte, E., 2017. Investment appraisal of cost-optimal and near-optimal pathways for the UK electricity sector transition to 2050. Appl. Energy 189, 89–109.
Mo, W., Nasiri, F., Eckelman, M.J., Zhang, Q., Zimmerman, J.B., 2010. Measuring the embodied energy in drinking water supply systems: a case study in the Great Lakes Region. Environ. Sci. Technol. 44 (24), 9516–9521.
Muñoz, I., Fernández-Alba, A.R., 2008. Reducing the environmental impacts of reverse osmosis desalination by using brackish groundwater resources. Water Res. 42 (3), 801–811.
Pomponi, F., Moncaster, A., 2018. Scrutinising embodied carbon in buildings: the next performance gap made manifest. Renew. Sustain. Energy Rev. 81, 2431–2442.
Rothausen, S.G., Conway, D., 2011. Greenhouse-gas emissions from energy use in the water sector. Nat. Clim. Change 1 (4), 210–219.
Singh, P., Kansal, A., 2018. Energy and GHG accounting for wastewater infrastructure. Resour. Conserv. Recycl. 128, 499–507.
Sinha, R., Lennartsson, M., Frostell, B., 2016. Environmental footprint assessment of building structures: a comparative study. Build. Environ. 104, 162–171.
Smyth, B.M., Davison, P., Brow, P., 2017. Carbon curves for the assessment of embodied carbon in the wastewater industry. Water Environ. J. 31 (1), 4–11.
Staffell, I., 2017. Measuring the progress and impacts of decarbonising British electricity. Energy Policy 102, 463–475.
Stokes, J., Horvath, A., 2010a. Life cycle assessment of urban water provision: tool and case study in California. J. Infrastruct. Syst. 17 (1), 15–24.
Stokes, J.R., Horvath, A., 2010b. Supply-chain environmental effects of wastewater utilities. Environ. Res. Lett. 5 (1), 014015.
Strutt, J., Wilson, S., Shorney-Darby, H., Shaw, A., Byers, A., 2008. Assessing the carbon footprint of water production. J. Am. Water Works Assoc. 100 (6), 80–91.
(UKGBC) UK Green Building Council, 2017. Delivering Low Carbon Infrastructure. London, UK. [ONLINE] Available at: https://www.ukgbc.org/wp-content/uploads/2017/09/Delivering-Low-Carbon-Infrastructure.pdf.
UKWIR (UK Water Industry Research), 2012. A Framework for Accounting for Embodied Carbon in Water Industry Assets. (Report Ref. No. 12/CL/01/15). UKWIR, London, UK.
Weidema, B.P., Bauer, C., Hischier, R., Mutel, C., Nemecek, T., Reinhard, J., Vadenbo, C.O., Wernet, G., 2013. Overview and Methodology: Data Quality Guideline for the Ecoinvent Database Version 3. Swiss Centre for Life Cycle Inventories.
Weidema, B.P., Wesnaes, M.S., 1996. Data quality management for life cycle inventories—an example of using data quality indicators. J. Clean. Prod. 4 (3–4), 167–174.
Xylem, 2013a. Flygt 3127.181 environment product declaration. http://gryphon.environdec.com/data/files/6/9530/epd466%20(3127_181).pdf, 18 March 2017.
Xylem, 2013b. Flygt 3085.900 environment product declaration. http://gryphon.environdec.com/data/files/6/9525/epd463%20(3085_900).pdf, 18 March 2017.
Xylem, 2013c. Flygt 3085.183 environment product declaration. http://gryphon.environdec.com/data/files/6/7230/epd62_3.1.pdf, 18 March 2017.